

Impact of plant diseases on faunal communities

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Abstract

Plant pathogens can have a major effect on vegetation floristics and structure. Effects include loss of plant species, decline in vegetation cover, and an increase in bare ground and the abundance of resistant plant species. These changes would be predicted to affect the faunal communities inhabiting infected habitats, but there have been few studies which examine the relationship between faunal abundance and composition and plant pathogens. This paper considers the potential affects of plant pathogens on faunal communities and reviews recent work on the effects of *Phytophthora cinnamomi* (cinnamon fungus) on mammals and invertebrates. Analysis of a number of disturbance factors in heathland and woodlands of south-eastern Australia have identified *P. cinnamomi* infection as being associated with low species richness, and low abundance of small mammals. Studies on populations of *Antechinus stuartii* (brown antechinus) in woodlands found that there were lower capture rates, and habitat utilisation was altered. The major contributing factor was alterations to vegetation structure, rather than food availability. In heathlands, species such as *Rattus lutreolus* (swamp rat), *Rattus fuscipes* (bush rat) and *Antechinus stuartii* were found to be less abundant in areas infected with *P. cinnamomi*.

Introduction

The habitat and microhabitat preferences exhibited by animal species are determined by a number of basic requirements such as the provision of adequate food resources, the presence of cover (protection from predators, suitable microclimate) and access to breeding, basking or roosting sites. Vegetation attributes make a major contribution to these habitat components. For example, understorey vegetation and litter provide important refuges and breeding areas for invertebrates. Omnivorous or herbivorous terrestrial lizards are dependent on vegetation as foraging sites, while some species require elevated sites for thermoregulation. Birds require vegetation for nest sites, some using tree cavities, others shrub or dense ground vegetation, and most species also depend on vegetation for food such as nectar, seeds, fruit or invertebrates. Small terrestrial mammals depend on vegetation for cover, food and protection. Tree bark, litter and woody debris, harbouring a diverse range of invertebrates, also provide foraging sites for insectivores.

Studies have provided evidence that small mammals exhibit microhabitat selection based on vegetation characteristics (Braithwaite & Gullan 1978, Braithwaite *et al.* 1978, Cockburn 1978, 1981, Fox & Fox 1981). Analyses of factors found that structural attributes of vegetation were important indicators for habitat preferences for some species (Barnett *et al.* 1978, Stewart 1979, Newsome & Catling 1979), while for others both floristic and structural characteristics were significant (Braithwaite & Gullan 1978, Braithwaite *et al.* 1978, Fox & Fox 1981). For example, a number of pseudomyine rodent species exhibit preferences for diverse

floristic communities, which are likely to be related to dietary requirements (Braithwaite & Gullan 1978, Cockburn 1978, 1981, Fox & Fox 1981, 1984). Species such as the omnivorous bush rat (*Rattus fuscipes*) respond mainly to structural factors while the herbivorous swamp rat (*Rattus lutreolus*) has a requirement for a sedge food resource (Braithwaite *et al.* 1978, Barnett *et al.* 1978).

Plant pathogens can have a major effect on vegetation communities. They may decrease the fitness of individual plants, alter the size and genetic structure of individual populations and thus the structure and diversity of whole plant communities (Burdon 1991). There may also be a range of effects from pathogens depending on the intensity of the pathogen and the conditions present at the time of infection. The consequences for animal habitats will also vary according to the intensity of pathogen effects and the percentage of resistant plants present. In some cases, marked changes to habitat conditions may occur if the pathogen affects all the vegetation present, or if there is total mortality of susceptible species. Indeed, some pathogens lead to almost complete loss of plants present, from the understorey to the canopy. In other situations only understorey vegetation may be affected. Thus, animals that rely on different habitat components would be affected differentially. For example, arboreal marsupials (*e.g.* folivores, nectarivores) and many bird species are likely to suffer as the consequence of canopy damage. Terrestrial mammals and invertebrates would be seriously affected if all understorey vegetation is eliminated. Furthermore, some animal species are reliant on particular plant species for their food. If those plant species are highly susceptible to a pathogen, then this will have important consequences for animal food resources.

In this paper we examine evidence and recent research on the impact of a particular plant pathogen, cinnamon fungus (*Phytophthora cinnamomi*) on fauna. We assess the affects of *P. cinnamomi* on vegetation together with consequential

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effects on animal populations, communities, habitats and diet. The role of *P. cinnamomi* as a disturbance regime in native communities is examined, and the long-term consequences of infestation and resilience of these communities is assessed.

Effects of *P. cinnamomi* on vegetation and animal habitats

The fungal pathogen *P. cinnamomi* causes extensive "dieback" of Australian native vegetation. It is widespread and occurs in forests ranging from the jarrah forests of Western Australia to the stringybark and silvertop ash forests of Victoria, and the tropical rainforests of Queensland. It also occurs in woodlands and heathland communities e.g. the Grampians and Brisbane Ranges, in Victoria and in Western Australia (Newhook & Podger 1972, Podger & Ashton 1970, Weste & Taylor 1971, Weste 1974, Wills 1993).

The pathogen has a wide host range, although its pathogenicity to different hosts varies, and its growth and distribution are influenced by temperature, soil type, nutrient status and water availability (Weste & Marks 1987, Marks & Smith 1991). Hence the expression of the disease varies from site to site. Studies of infected vegetation have shown that *P. cinnamomi* has a major effect on the understorey, and up to 60% of the plant species present have been eliminated after infection (Weste 1974, Dawson *et al.* 1985, Kennedy & Weste 1986, Wills 1993). Infection results in decreases in seedling regeneration, species diversity and the population density of species (Weste 1986, Dawson *et al.* 1985, Kennedy & Weste 1986), but there is often an increase in the frequency and cover of field resistant monocotyledons (Dawson *et al.* 1985, Weste 1986, Kennedy & Weste 1986). In severely affected areas, the understorey community has been changed from diverse sclerophyll to a sedge-dominated understorey, due to the resistance of sedges to *P. cinnamomi*. In some situations large areas have become denuded of vegetation, often resulting in severe erosion (Kennedy & Weste 1986).

The changes in vegetation communities associated with *P. cinnamomi* infection would be predicted to affect the fauna present (Table 1). Decreased canopy cover could potentially affect nesting birds and arboreal marsupials, while simplification of the understorey vegetation may affect sources of seed, nectar and pollen available for mammals, birds and insects. Indeed, species such as the honey possum (*Tarsipes rostratus*), which rely entirely on a specialized diet of pollen and nectar mainly from highly susceptible proteaceous species (e.g. Banksias), are likely to be severely affected by *P. cinnamomi* (Friend 1992).

Alterations to understorey vegetation and litter could alter habitat conditions for invertebrates. Thus, one may expect changes in the abundance and composition of invertebrate communities which in turn influence the diets of insectivorous animals such as small dasyurid marsupials. Dibblers (*Parantechinus apicalis*), for example, were rediscovered in 1967 at Cheyne Beach on the south coast of Western Australia (Morcombe 1967), and shown to forage on invertebrates which inhabited the wildflowers and deep litter layer characteristic of the site. Since that time, this habitat has been severely affected by both *P. cinnamomi* and the canker fungus *Diplodonia* spp., and the status of *P. apicalis* in this area is uncertain. Like the honey possum, the dabbler is potentially under great threat from *P. cinnamomi* and other pathogens.

Table 1
Predicted effects on fauna due to the presence of *P. cinnamomi* (cinnamon fungus) in vegetation communities.

Effects on vegetation	Effects on fauna
1. Loss of susceptible plant species in the understorey, midstorey	a) Direct loss of food sources e.g. seeds, nectar, pollen. b) Indirect loss of food sources e.g. invertebrates.
2. Decline in plant species' richness and diversity	a) Loss of food for species that prefer floristically rich vegetation. b) Loss of seasonal food availability.
3. Decrease in plant cover, increase in bare ground, erosion.	a) Loss of habitat for species dependant on thick ground cover. b) Increased predation risk. c) Changes to microclimate conditions.
4. Decrease in canopy cover	a) Loss of food for arboreal species. b) Loss of habitat for arboreal species.
5. Decrease in litter fall.	a) Decline in litter invertebrates (dry conditions) b) Decline in invertebrate food sources for insectivores.
6. Post infection increase in frequency of field resistant plant species e.g. sedges.	Increase in food for specialist herbivores

Species such as the pseudomyine rodents have been shown to be dependent on floristically-diverse understorey (Cockburn 1978, Cockburn *et al.* 1981, Fox & Fox 1984) and are potentially endangered by the simplification of diverse sclerophyll communities resulting from *P. cinnamomi* infection. By contrast, animals with relatively generalized diets that require dense, low vegetation for shelter may prefer habitat that has been infected by *P. cinnamomi*, if there has been a consequent increase in the cover of field resistant monocotyledons. In severely affected areas, decreases in plant cover could lead to loss of habitat or a decline in the carrying capacity for these animals.

Effects of *P. cinnamomi* on animal populations

Although there have been a substantial number of studies on the effects of *P. cinnamomi* on vegetation (see review by Weste & Marks 1987), there has been little work investigating the less direct effects of the pathogen on faunal populations and communities.

Two studies have compared invertebrate communities in healthy and infected jarrah forests in Western Australia. Postle *et al.* (1986) found that infected jarrah forest had 48% less litterfall, and a standing biomass of leaf litter 8.4% of healthy forest, although these differences were not tested statistically. Numbers of soil and litter invertebrates were generally lower in diseased forest, but there were variations between seasons and between taxa for both the soil and litter components. Nichols & Burrows (1985) recorded lower

numbers of invertebrate species and individuals in a diseased forest, as well as fewer trees and shrubs and lower mean litter cover values. Variations in abundance of invertebrate taxa were also observed between an uninfected forrest (*i.e.* $n=1$) and an infected ($n=1$) forest depending upon the habitat requirements of each taxon. For example, Dermaptera which require dense litter cover were only recorded in healthy forest, whereas the majority of ant species were found in both diseased and healthy forests, although the total abundance of ants was lower in diseased forest. The low number of sites in these studies makes statistically-testable conclusions impossible.

Reptiles and frogs have been surveyed in both healthy and diseased jarrah forest. Diseased forest supported lower numbers of species and lower abundances than healthy forest (Nichols & Bamford 1985). Some species however, were more abundant in diseased forest (*e.g.* *Pogonaminor* and *Cryptoblepharus plagiocephalus*), perhaps reflecting the increased insolation on elevated surfaces (*e.g.* logs) which these species use for basking and foraging (Nichols & Bamford 1985; Wilson & Knowles 1988). Again, low site numbers precluded statistical testing of the data.

There is some evidence that there are differences in the avifauna present in diseased and uninfected forests. Nichols & Watkins (1984) described a dieback-affected forest that had low bird species richness and abundance compared with healthy sites. At another dieback site, however, the bird density and species richness were comparable to those in healthy forest. Some species were recorded in higher densities in diseased forest, while other species were absent, resulting in different bird species composition in the two forest types.

In a study of small mammal communities in heathy woodland and heathlands, Wilson (1990) and Wilson *et al.* (1990) found the percentage of vegetation modified by *P. cinnamomi* to be a significant variable in explaining small mammal diversity and density. Further studies by Newell & Wilson (1993) in heathy communities of the Brisbane Ranges National Park, Victoria, found the abundance of *Antechinus stuartii* to be lower in *P. cinnamomi* affected areas. Vegetation volume was significantly lower between 0 and 60 cm, in diseased areas and the abundance of *A. stuartii* was significantly correlated with this variable. The change in structure in diseased areas was predominantly related to the loss of the austral grass-tree (*Xanthorrhoea australis*). Whether this loss affected *A. stuartii* through reduced cover, or altered food availability was unclear. Studies in coastal heathland at Anglesea, in southern Victoria, found that several small mammal species *e.g.* *R. lutreolus* (swamp rat), *R. fuscipes* (bush rat), and *A. stuartii* (brown antechinus) were less abundant in diseased heathland than in healthy stands (Laidlaw & Wilson unpublished data). Mean species richness of small mammal communities was also lower at infected sites.

The utilization of habitat by the dasyurid marsupial *A. stuartii* in *P. cinnamomi* infected and non-infected areas have been investigated at several sites in the Brisbane Ranges, Victoria. *A. stuartii* was found to forage almost exclusively at ground level, and frequently used nest sites located at ground level in large *X. australis* plants (Newell 1994). The movement and home range of *A. stuartii* were investigated

using trapping and radiotelemetry. Home ranges displayed a high degree of overlap with areas that were uninfected with *P. cinnamomi* and animals actively selected uninfected habitat, and avoided areas infected with *P. cinnamomi* (Newell 1994). Individuals occasionally crossed bare, long-term infected areas to enter other uninfected habitat. These results suggest that *A. stuartii* relied heavily on vegetation cover. The effect of *P. cinnamomi* on the dietary items of *A. stuartii* was also investigated (Newell 1994). There was no relationship between invertebrate abundance, and the capture rate of *A. stuartii* in infected or uninfected areas.

The above studies provide evidence that the modification of habitat due to the presence of *P. cinnamomi* can lead to declines in the overall abundance of fauna. There is evidence that these changes also result in reduction in species richness, and/or diversity. Changes in the abundance of individual taxa, may depend on their habitat requirements, with some taxa increasing and others declining. Some of the studies outlined were limited in design, and mainly addressed other environmental disturbances such as mining. There is evidence of altered utilization of habitat by *A. stuartii* due to the presence of *P. cinnamomi*. Further work needs to be undertaken to establish the relationships between the modification of habitat components and changes in faunal communities due to the affects of *P. cinnamomi*.

P. cinnamomi, disturbance and resilience

Australian vegetation and fauna communities are well adapted to disturbance factors such as fire. In areas where fire frequency is high, communities often exhibit high degrees of resilience (or more specifically, elasticity; *sensu* Waltman 1986) following disturbance. This resilience is likely to be a consequence of the evolution of adaptive features over long periods of exposure to disturbance regimes. The presence of *P. cinnamomi* may also be considered a component of a disturbance regime. However, there is evidence that *P. cinnamomi* has recently been introduced to Australia (Weste & Marks 1974, 1987) and native plant and animal communities have not been exposed to the regime for a long period of (evolutionary) time. Therefore these communities are unlikely to have adapted to the presence of *P. cinnamomi*.

Compared with the relatively ephemeral effects of fire, *P. cinnamomi* may have a severe and long-lasting impact on plant and animal communities. Infection generally results in removal of a wide range of susceptible species and leads to simplification of the understorey. Furthermore, the pathogen is capable of remaining in the soil following infection, so may re-infect vegetation or infect new vegetation. Secondary plant succession after fire normally results in the re-establishment of the original floristic community, following an initial floristic composition model (Noble & Slayter 1981). There is evidence, however, that susceptible plant species rarely recolonise after *P. cinnamomi* infection (Weste & Marks 1987). The only evidence of recolonisation is where the austral grass-tree (*Xanthorrhoea australis*) has been recorded, at least 20 years after infection (Dawson *et al.* 1985, Weste 1993). Given large alterations to vegetation species composition after *P. cinnamomi* infestation, it is difficult to predict regeneration of the original vegetation community (Grubb & Hopkins 1986).

There is often substantial recolonisation by small mammals after fire (Fox 1982; Fox & McKay 1981, Newsome *et al.* 1975, Wilson *et al.* 1990). Recolonisation rates depend on factors such as the fire regime, the regeneration of vegetation and sources of recolonisers (*i.e.* unburnt patches). Some species enter the succession early (2-4 years) due to their preference for diverse young vegetation, while others may take up to ten years to return (Fox 1982, Wilson *et al.* 1990). Successional changes of vegetation communities following *P. cinnamomi* infestation are yet to be established. The resultant community is likely to differ substantially from the original vegetation community. It is presently difficult to accurately determine animal succession patterns; however, one would expect a very slow recolonisation process resulting in a different faunal community.

Conclusions

It is clear from the above review that *P. cinnamomi* has the potential to severely influence the abundance and composition of many faunal communities. These effects are largely indirect, resulting from changes in plant species richness and composition, and from alterations to the structural components of the habitat. Although data are limited, there is evidence that a broad range of taxa are affected including small mammals, reptiles, birds and invertebrates. The effect on individual taxa will depend on that species' requirements for food and shelter, and its reproductive and foraging strategies. It may be predicted that some generalist species which require relatively open habitats may be favoured, while more specialized species (especially those with restricted diets which inhabit dense species-rich shrublands) would markedly decline in the dieback-affected areas. Similar principles have been shown to apply in assessing the impact of fire on small vertebrates (Friend 1993), and open the way for predictive modelling of the impact of plant diseases on faunal populations.

In tandem with these broad approaches to develop a classification of tolerant and sensitive faunal species (as has been done for many plant species; Wills 1993), there needs to be more detailed studies of movement patterns and resource utilization for the most sensitive species, particularly those which are regarded as rare or endangered. Although detailed habitat utilization work has been carried out on *A. stuartii* in relation to infection by *P. cinnamomi* (Newell 1994), there has been little such work on other potentially sensitive species like *T. rostratus*, *P. apicalis* and the western pygmy possum (*Cercartetus concinnus*).

Unique and threatened communities also need to be identified, and measures enforced to assure their long-term protection from plant diseases. Such work is now underway in Western Australia, where a GIS-based decision support system is being developed to monitor and manage *Phytophthora*-sensitive taxa and communities, and experiments are being carried out to evaluate the efficacy of aerial application of phosphonate to control *Phytophthora* in native plant communities. It is only through application of such multidisciplinary studies that we can begin to understand the processes by which plant diseases influence the structure and composition of plant and animal communities.

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